

Microbial Degradation and Humification of the Lawn Care Pesticide 2,4-Dichlorophenoxyacetic Acid during the Composting of Yard Trimmings

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The fate of the widely used lawn care herbicide 2,4-dichlorophenoxyacetic acid (2,4-D) during the composting of yard trimmings consisting of primarily leaves and grass is an important unexplored question. In this study, we determined the extent of 2,4-D mineralization, incorporation into humic matter, volatilization, and sorption during the composting of yard trimmings. Yard trimmings (2:1 [wt/wt] leaves-grass) were amended with ^{14}C -ring-labeled 2,4-D (17 mg/kg of dry weight) and composted in a temperature-controlled laboratory scale compost system. During composting, thermophilic microbes were numerically dominant, reaching a maximum of $2 \times 10^{11}/\text{g}$. At the end of composting, 46% of the organic matter (OM) present in the yard trimmings was lost and the compost was stable, with an oxygen uptake rate of 0.09 mg of O_2 per g of OM per h, and was well humified (humification index, 0.39). Mineralization of the OM temporally paralleled mineralization of 2,4-D. In the final compost, 47% of the added 2,4-D carbon was mineralized, about 23% was complexed with high-molecular-weight humic acids, and about 20% was not extractable (humic fraction). Less than 1% of the added ^{14}C was present in water expressed from the finished compost, suggesting a low potential for leaching of 2,4-D. Very little volatilization of 2,4-D occurred during composting. It is of interest that our results indicate active mineralization of 2,4-D at composting temperatures of 60°C because microbial 2,4-D degradation at thermophilic temperatures has not been previously documented.

A number of states in the United States have proposed bans on land filling with and incineration of yard trimmings (yard wastes), leaving composting as the primary recycling option in many communities. During the composting of yard trimmings, lignocellulosic materials are in part mineralized and in part converted to humic substances (19). Finished composts resemble high-organic-matter (OM) soils in their physical and chemical properties (7, 12, 21, 24).

Over 30,000 tons of pesticides is used annually on lawns and gardens in the United States (1, 27). Although there have been many studies on the degradation of pesticides in soils, very little research has been done on the fate of these compounds during the composting of yard trimmings (7, 15, 22). This knowledge is important, since many of the pesticides applied to leaves and grass are toxic and mutagenic. Furthermore, undegraded pesticides could potentially be concentrated during composting, as degradable OM is mineralized to carbon dioxide and water. Hence, there is a growing interest in the fate of pesticides during composting. There are a number of potential fates for pesticides during composting. These include mineralization to carbon dioxide, conversion to humic matter, incorporation into microbial biomass, adsorption, biotransformation, volatilization, and leaching.

2,4-Dichlorophenoxyacetic acid (2,4-D) is the most commonly used residential lawn care herbicide (1). Previous studies have shown that turf grass cut within a week of a standard application of 2,4-D contains this compound at 26 to 183 ppm (3). On the other hand, samples of incoming grass clippings at yard waste composting facilities have been shown to contain 2,4-D at up to 5.5 ppm (8). Mineralization of 2,4-D has been shown to occur in agricultural soils (13, 16, 17), as well as by

pure cultures of bacteria (6, 9, 23) and fungi (29). However, there have been no studies on the extent of degradation and the fate of 2,4-D during the composting of yard trimmings. Moreover, no pure cultures are known to degrade 2,4-D at temperatures above 45°C (25). In this report, we describe our findings on the fate of 2,4-D during composting of yard trimmings and demonstrate mineralization of 2,4-D under thermophilic conditions.

MATERIALS AND METHODS

Feed stock. Yard trimmings consisting of leaves and grass (2:1 on a dry-weight basis) were collected at a municipal composting site in Lansing, Mich., dried at 37°C to approximately 7% moisture, and then ground in a Viking C346554 electric hammer mill (Viking Mfg. Co., Jackson, Mich.) to allow passage through a 3-mm screen. The yard trimmings were sprayed with a sterile solution containing U-ring- ^{14}C -2,4-D and unlabeled 2,4-D (sodium salt) to give a total concentration of 17 mg kg^{-1} (17 ppm) on a dry-weight basis. The specific activity of the ^{14}C -2,4-D used was 0.267 mCi/mmol, and its chemical purity was >98% on the basis of high-performance liquid chromatography analysis provided by the supplier (Sigma, St. Louis, Mo.). Approximately 200 g (dry weight) of the 2,4-D-amended yard trimmings was added to each of three replicate composters. The total amount of ^{14}C added to each composter was approximately 4.0 μCi . Distilled water was added to each composter to adjust the moisture content to 60%, which is comparable to that in leaf and grass mixtures composted at large-scale facilities (18, 24). No exogenous inocula were added.

Compost reactor system. The composting system used in this study was a modification of that described previously (19). Briefly, the compost chambers consisted of rubber-stoppered 2-liter wide-mouth glass jars with two plastic screens (1-cm mesh opening) and one polyethylene screen (1-mm mesh opening) serving as a false floor (Fig. 1). Aeration was provided through an opening in the bottom of each jar. The air supply was stripped of CO_2 by passage through a packed bed column filled with 1 liter of 5 N NaOH. To avoid moisture loss, the air entering the compost chamber was humidified at the incubation temperature. The air flow rate into each composter was approximately twice the rate needed to supply adequate oxygen for microbial respiration as estimated from the CO_2 evolution rates determined in preliminary experiments. The air flow rates, measured with a rotameter (Aalborg Instruments, Monsey, N.Y.), were 100 ml/min during the first 15 days and 50 ml/min thereafter. The composting system was

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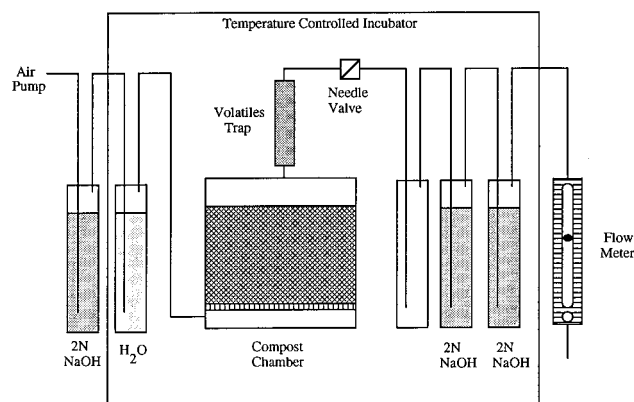


FIG. 1. Schematic of the laboratory scale compost system used to determine the fate of ^{14}C -2,4-D. The volume of the reactor was 2 liters. Air was passed sequentially through 2 N NaOH to strip it of CO_2 and then into an incubator, where it was humidified by passage through deionized water. The exit gas from the compost chamber entered a two-stage polyurethane foam trap and then passed through two flasks, each containing 50 ml of 2 N NaOH to trap the $^{14}\text{CO}_2$ produced.

contained in a 10632M-1 temperature-controlled incubator (Lunaire Environmental, Inc., Williamsport, Pa.).

The exhaust gas from the compost chamber was passed through a series of two polyurethane foam plugs (3 by 2 cm), which had previously been cleaned by Soxhlet extraction with hexane, to trap any volatile organics generated from 2,4-D metabolism (22). The polyurethane plugs were removed on days 5, 10, 18, 27, 40, and 54 and were extracted with ether. An aliquot of the extracts was mixed with 18 ml of Safety Solve scintillation cocktail (Research Products International, Mount Prospect, Ill.), and the radioactivity was determined with a liquid scintillation analyzer (TriCarb 1500; Packard Instrument Co., Downers Grove, Ill.). Almost all of the adsorbed radioactivity was found in the first foam plug in the series. The exit gas was then passed through two successive CO_2 traps, each containing 50 ml of 3 M NaOH. Aliquots (0.5 to 1 ml) of the NaOH solutions with trapped $^{14}\text{CO}_2$ were removed periodically, diluted with 2 ml of deionized water, mixed with 18 ml of Safety Solve scintillation cocktail, and counted with a scintillation analyzer as described above. To ensure that the ^{14}C in the NaOH traps was due to $^{14}\text{CO}_2$, a BaCl precipitation step was included (28). Between 98 and 99.5% of the counts present in the sodium hydroxide traps were barium chloride precipitable, indicating that almost all of the trapped counts in the NaOH solutions were due to $^{14}\text{CO}_2$ and not due to volatilized 2,4-D or other organics derived from 2,4-D which may have escaped the polyurethane foam traps. Total CO_2 evolution was determined by titration of the NaOH solution to a phenolphthalein endpoint. Control tubes (no compost in the circuit) were also titrated. Background radioactivity was measured by using samples from control composters without added ^{14}C -2,4-D.

The temperature program used for the composting system was designed to mimic the temperatures typically found during large-scale windrow composting of a 2:1 leaf-grass mixture where temperatures reach a peak of 60 to 70°C after 8 to 10 days and remain between 50 and 60°C through day 80 (18). The temperature of the incubator was programmed to rise by 5°C/day and was held at 60°C after day 8.

Compost analyses. Samples (20 g [wet weight]) for analysis were removed from three identical compost reactors on days 0, 5, 10, 16, 22, 31, and 49. Approximately 10 g of the sample was dried at 105°C and ashed at 550°C to determine moisture content and OM content, respectively. The OM refers to the difference between ash and dry-weight values. The pH was determined by using

diluted compost samples (1:50) as described previously (4). Compost stability was measured as the oxygen uptake rate (milligram of O_2 gram of $\text{OM}^{-1} \text{h}^{-1}$) at 37°C by the method of Iannotti-Frost et al. (11). The humic and fulvic acid content was determined as described by Ciavatta et al. (5). The carbon content of the humic acid, fulvic acid, and nonhumic fractions was determined by dichromate oxidation in accordance with the manufacturer's instructions (Hach and Co., Loveland, Colo.).

The total ^{14}C in the compost was determined by oxidizing 500-mg samples of compost at 900°C in a biological materials oxidizer, model OX300 (R. J. Harvey, Instrument Corp., Hillsdale, N.J.). The samples were combusted for 4 min at an oxygen flow rate of 350 ml/min. The evolved $^{14}\text{CO}_2$ was trapped in a trapping solution consisting of a 2:1 mixture of Carbosorb (Research Products International, Mount Prospect, Ill.) and Safety Solve scintillation cocktail and counted as described above. The $^{14}\text{CO}_2$ trapping efficiency was 97.5%. To determine the distribution of radioactivity remaining in the compost, 5-g samples were extracted successively with ether, 0.15 M phosphoric acid, and 0.15 M NaOH-0.15 M $\text{Na}_4\text{P}_2\text{O}_7$ (NaOH-extractable fraction) by the procedure of McCall et al. (17). The latter fraction was further separated into humic acid, fulvic acid, and nonhumic fractions as follows. Humic acids were precipitated by adjusting the pH to 1.5, and the supernatant was separated into fulvic acid and nonhumic fractions by adsorption of fulvic acid on solid polyvinylpyrrolidone resin (Sigma) (5). The radioactivity remaining after extractions was determined by oxidizing the insoluble humin in a biological materials oxidizer and trapping and counting the evolved $^{14}\text{CO}_2$.

To determine the potential leachability of 2,4-D during composting and the amount of 2,4-D sorbed to OM, 10 g of the ^{14}C -2,4-D-amended compost was pressed in a soil press until no more water was expressed (10). The compost water was collected and centrifuged, and the ^{14}C content of the supernatant liquid was determined with a liquid scintillation counter as described above.

Molecular weight (MW) distribution of ^{14}C associated with humic acids. To determine the MW distribution of the humic acids containing ^{14}C from 2,4-D, the NaOH-extractable fraction was concentrated 10-fold by evaporation under a vacuum and applied to a gel permeation chromatography column (P-2 Biogel; Bio-Rad Laboratories, Richmond, Calif.) which excludes compounds with MWs of $>1,600$. Pure ^{14}C -2,4-D in the NaOH extract from day 50 compost (control) was also fractionated on a similar P-2 Biogel column. The total volume of each column was 40.7 ml, the exclusion volume was 17.2 ml, and the flow rate was 0.5 ml/min. The columns were eluted with pH 9.0 Tris buffer, and the amounts of radioactivity in various fractions were determined by scintillation counting as described above.

Media and culture conditions. To estimate the total number of heterotrophic bacteria, most-probable-number (MPN) determinations were done by adding 10-g samples of compost to 90 ml of sterile 0.1 M phosphate buffer, homogenizing the mixture for 5 min in a Sorvall tissue homogenizer, and adding aliquots (20 μl) of this solution to 180 μl of nutrient broth in the first row of a 96-well culture plate. The aliquots were then serially diluted to give compost dilutions of 2×10^{-3} to 2×10^{-12} g per well. There were eight tubes per dilution (equivalent to an eight-tube MPN). The plates were incubated at room temperature and at 55°C for 48 h. The A_{630} in each well was then measured with a plate reader (Microplate Autoreader EL311; Biotek Instruments). Positive wells showing an A_{630} of greater than 0.15 compared with an uninoculated control well were scored, and the MPN was calculated as described previously (14).

RESULTS

Compost characterization. Yard trimmings amended with ^{14}C -2,4-D were composted for 50 days. During this period, the OM content decreased from 68 to 53.5% (grams of OM per gram of dry weight), which represents a loss of 45% (Table 1). The pH rose from 6.9 initially to 8.3 by day 10 and remained essentially stable thereafter. Compost stability, measured as the oxygen uptake rate, was 0.09 mg of O_2 g of $\text{OM}^{-1} \text{h}^{-1}$ in the finished compost. By comparison, the initial feedstock ex-

TABLE 1. Characteristics of yard trimmings before and after composting^a

Time of measurement ^b	% Carbon	% Nitrogen ^c	C/N ratio	% OM	O_2 uptake rate ^d	pH	Humic and fulvic acid ^e	Nonhumic ^e	HI ^f
Before composting	43.2	1.44	30	68 \pm 0.2	3.5 \pm 0.2	6.9 \pm 0.3	35 \pm 3	36 \pm 2	1.0 \pm 0.2
After composting	32.8	1.76	19	53.5 \pm 0.6	0.09 \pm 0.03	8.3 \pm 0.2	60 \pm 1	22 \pm 4	0.37 \pm 0.07

^a All of the values shown were determined on a dry-weight basis. Means for triplicate composters \pm 1 standard deviation are shown.

^b Initial and final samples were taken on days 0 and 50 of composting, respectively.

^c The Kjeldahl procedure was used for determining total nitrogen.

^d Expressed in milligrams of O_2 gram of $\text{OM}^{-1} \text{hour}^{-1}$.

^e Expressed in milligrams of C gram of OM^{-1} .

^f HI, humification index.

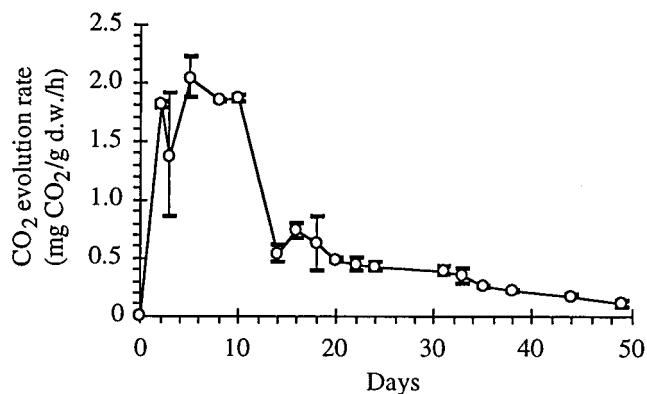


FIG. 2. Total CO₂ evolution during the composting of yard trimmings. The values presented are means \pm 1 standard deviation for triplicate composters. d.w., dry weight.

hibited an O₂ uptake rate of 3.49 mg of O₂ g of OM⁻¹ h⁻¹. Composts with O₂ uptake rates of <0.1 mg of O₂ g of OM⁻¹ h⁻¹ are generally considered stable (11, 19). The CO₂ evolution rate during composting increased steadily for the first 8 days and reached a maximum of 2.0 mg of CO₂ g of dry weight⁻¹ h⁻¹ between days 3 and 8 and then gradually decreased to below 0.5 mg of CO₂ g of dry weight⁻¹ h⁻¹ after day 40 (Fig. 2). The amount of carbon present in the nonhumic fraction decreased by almost half, while that in the combined humic and fulvic acid fractions nearly doubled during composting (Table 1). The humification index (ratio of nonhumic extractables to humified extractables), which is often used as an indicator of compost maturity, decreased from 1.0 initially to 0.37 in the final composts, indicating that the final composts were well humified (5).

Total MPN in the 2,4-D-amended compost increased substantially during composting. Initially there were 9×10^7 mesophiles and 5×10^5 thermophiles (data not shown). As the composting progressed and the temperature increased, thermophiles replaced mesophiles as the numerically dominant group. The MPN of thermophiles reached a maximum of 2×10^{11} /g of dry weight on day 20 and then decreased to 2×10^{10} by day 50.

Fate of ¹⁴C-2,4-D during yard trimming composting. About 27% of the 2,4-D was mineralized after 10 days of composting (Fig. 3). After 50 days of composting, $47\% \pm 6\%$ of the initial

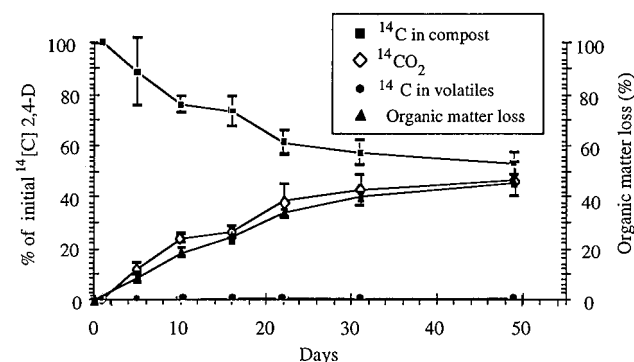


FIG. 3. Amount of ¹⁴C-2,4-D mineralized to ¹⁴CO₂, volatilized, and remaining in compost and percentage of total OM lost during composting of yard trimmings amended with 2,4-D at 17 ppm (wt/wt). The values presented are means \pm 1 standard deviation for triplicate composters corrected for percent recovery as described in Table 2, footnote d.

TABLE 2. Distribution of ¹⁴C from ¹⁴C-2,4-D in various fractions of initial feedstock and final compost

Component	% Distribution of ¹⁴ C ^a		
	Initial	Final	Autoclaved
¹⁴ CO ₂	0	47.7 \pm 6.7	0
Ether-extractable fraction	92.4 \pm 2.6	1.3 \pm 0.5	54.4 \pm 3.3
Dilute-acid-extractable fraction	3.4 \pm 2.2	2.5 \pm 0.5	1.6 \pm 0.1
NaOH-extractable fractions			
Total ^b	3.7 \pm 0.7	29.1 \pm 2.4	4.4 \pm 1.8
Fulvic acid	0.2 \pm 0.1	4.1 \pm 0.1	ND ^c
Humic acid	3.4 \pm 0.4	22.9 \pm 1.9	ND
Nonhumic	0.1 \pm 0.2	2.1 \pm 0.4	ND
Nonextractable humin	0.5 \pm 0.1	19.5 \pm 4.6	39.6 \pm 2.8
% Recovery ^d	99 \pm 5	97 \pm 10	97 \pm 1

^a Values are averages for triplicate composters \pm 1 standard deviation adjusted for percent recovery. The initial samples were taken on day zero of composting. Final and autoclaved (abiotic control) samples, were taken on day 50.

^b Extracted with 0.15 M NaOH–0.15 M Na-PP_i. See Materials and Methods for details.

^c ND, not determined.

^d Percent recovery was calculated as the mean \pm 1 standard deviation of the percentages of ¹⁴C recovered from triplicate composters.

¹⁴C-2,4-D was mineralized to ¹⁴CO₂. During this same period, $46\% \pm 2\%$ of the total carbon was converted to CO₂. Mineralization of 2,4-D temporally paralleled the mineralization of total compost OM to CO₂ (Fig. 3). Continued mineralization of 2,4-D even after the temperature had risen to 60°C indicated that microbes present in the compost are capable of mineralizing 2,4-D under thermophilic conditions. In control composters containing autoclaved feedstock, no 2,4-D mineralization was observed. A negligible amount of 2,4-D was volatilized during composting. The total amount of ¹⁴C recovered from the polyurethane traps after 50 days of composting was $0.4\% \pm 0.2\%$ of the initial ¹⁴C added. The overall percent recovery of ¹⁴C in the NaOH traps and the polyurethane foam traps and that remaining in the compost (amount of ¹⁴C measured in day zero compost divided by the amount of ¹⁴C recovered from day 50 compost) was $97\% \pm 10\%$.

Distribution of ¹⁴C-2,4-D during composting. Composting resulted in a dramatic change in the extractability of the added ¹⁴C-2,4-D carbon. After the first day of composting, $92\% \pm 3\%$ of the initial ¹⁴C was ether extractable but only $1.3\% \pm 0.5\%$ of the initial ¹⁴C was extractable after 50 days of composting (Table 2). Most of the unmineralized 2,4-D in the final compost was present in the humic acid and humin (unextractable) fractions (Table 2). Autoclaved feedstock amended with ¹⁴C-2,4-D and incubated for 50 days was also fractionated to determine the extent of abiotic 2,4-D transformation (Table 2).

To determine the MW distribution of the ¹⁴C-containing products in the humic acid fraction of the final compost, the NaOH fraction was fractionated by gel permeation chromatography. Approximately 70% of the ¹⁴C present in the compost extract eluted with the void volume of the column, indicating that these compounds had MWs greater than or equal to 1,600 (Fig. 4). Approximately 30% of the ¹⁴C in the extract had a MW of 700. There was no ¹⁴C at elution volumes corresponding to a MW of less than 500. Since the MW of 2,4-D is 221, these results indicate that part of the added 2,4-D was converted to high-MW humic acids during composting.

Sorption of 2,4-D during composting. To determine the

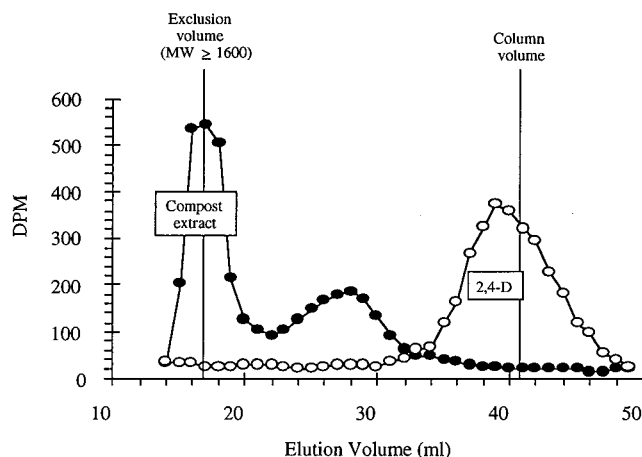


FIG. 4. Gel chromatography of the NaOH-extractable fraction of ^{14}C -2,4-D-amended compost (see Table 2) and of standard ^{14}C -2,4-D.

amount of 2,4-D sorbed to compost OM, a soil press was used to express the compost water and the radioactivity in the expressed water was determined and compared with the total amount present in the compost. On day 1, the concentration of ^{14}C -2,4-D and 2,4-D metabolites in the water fraction of the compost accounted for 25% of the total ^{14}C -2,4-D added to the compost (Fig. 5). Ten days later, this had declined to 8%, and after 20 days of composting, the expressed compost water contained less than 3% of the ^{14}C remaining in the compost. This would be equivalent to a 2,4-D concentration of 0.3 ± 0.2 mg/liter if all of the radioactivity in the expressed liquid was due to 2,4-D.

Mineralization of high levels of 2,4-D under thermophilic conditions. The results presented above indicated for the first time that microbial degradation of 2,4-D occurs under thermophilic conditions. Therefore, we investigated this aspect further by adding high levels of 2,4-D (2,500 ppm) and by maintaining the compost temperatures continuously at 55°C beginning from day zero. The results showed that under these conditions, 18% of the ^{14}C -2,4-D (i.e., 450 ppm) was mineralized in 50 days (Fig. 6). These data clearly indicated that microbial populations in the compost were capable of degrading high levels of 2,4-D under thermophilic conditions.

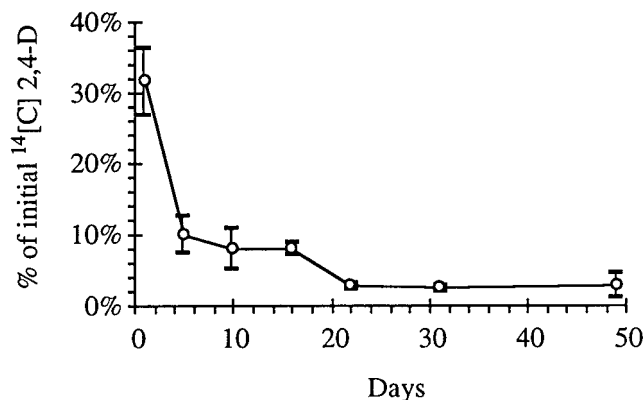


FIG. 5. Soluble ^{14}C present in water expressed from ^{14}C -2,4-D-amended yard trimmings compost. The values presented are means ± 1 standard deviation for triplicate composters.

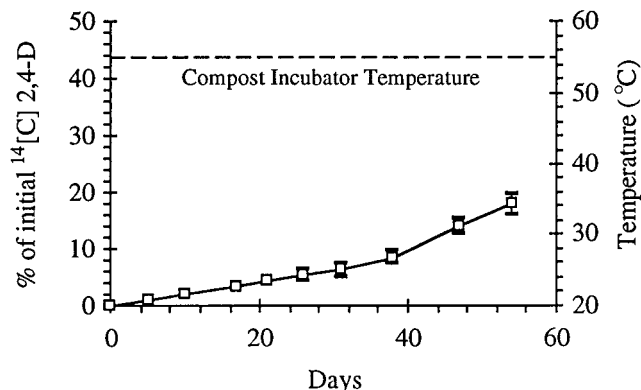


FIG. 6. Amount of ^{14}C -2,4-D mineralized at 55°C during the composting of yard trimmings amended with high levels of 2,4-D. The compost feedstock was amended with 2,4-D at 2,500 ppm (wt/wt), and the compost temperature was maintained at 55°C from day zero until the end of the composting. The values presented are means ± 1 standard deviation for triplicate measurements from one composter.

DISCUSSION

Composting is a dynamic process in which OM is converted to CO_2 , water, biomass, and humus by microbial action. On a wet-weight basis, a 30% weight loss was observed during the composting of yard trimmings in the laboratory scale system. The final compost product was well humified and stable (Table 1), and the initial and final pHs, moisture contents, OM contents, C-N ratios, and humification indices were all similar to those observed during field scale composting of yard trimmings (18, 24). The rate of OM loss in the laboratory scale system was faster than that generally observed under field scale conditions, probably because of more sufficient aeration conditions and the relatively smaller particle size of the leaves and grass used in the laboratory system.

The MPN of heterotrophic microorganisms observed during the composting of yard trimmings was $2 \times 10^{11}/\text{g}$. This was more than 2 orders of magnitude greater than the 10^6 to 10^8 bacteria per g typically observed in soils (21). Greater MPNs may be due to the relatively greater amount of readily degradable organic substrates in compost than in soils and the optimal environmental conditions present in the compost reactors. The MPNs determined here were also greater than those determined previously in composts where between 10^7 and 10^{10} CFU/g were observed (26). However, this difference may be due to the difference between the MPN method used by us and the plate counting methods used by previous investigators (26). Determination of viable organisms by plate counting has been shown to underestimate the number of organisms present in compost (2). Our MPN estimates of the numbers of heterotrophic microorganisms are in agreement with values determined by direct counting using 2,4-diamidinophenylindole (DAPI) staining (2).

The results of this study show that almost 50% of the ^{14}C -2,4-D carbon is mineralized, 23% is converted to humic acids, and 19.5% is unextractable after composting. In contrast to this, an abiotic control showed no detectable mineralization of 2,4-D and little conversion to humic acids, but almost 40% of the radioactivity was unextractable (Table 2). These results suggest that humification of 2,4-D during composting is primarily a biological process and that the unextractability of 2,4-D under abiotic conditions was primarily a physical-chemical process.

Leachates from large-scale yard waste composting facilities

are of concern, since these may contain pesticide and other residues which could contaminate ground water. The water- and ether-extractable ^{14}C from ^{14}C -2,4-D decreased dramatically during composting. Most of the 2,4-D was ether extractable on day 1, but after 50 days of composting, only 1.3% of the ^{14}C was ether extractable, even though more than 50% of the added ^{14}C was still present in the compost. Our results show that most of the 2,4-D and 2,4-D metabolites present in the compost are sorbed and therefore are not readily leachable during composting (Fig. 5).

The results indicate that the fate of 2,4-D during composting is similar to the fate of 2,4-D in soils. McCall et al. (17) observed that of the 1 ppm of ^{14}C -2,4-D added to six different soils, between 60 and 70% was mineralized after 60 days of incubation. This compares to the 48% mineralization of ^{14}C -2,4-D that we observed after 50 days of composting. In both soils (17) and compost, the amount of 2,4-D carbon which was ether extractable declined from nearly 100% initially to less than 5% after 50 days. The radioactivity in the NaOH-extractable and humin fractions accounted for 30 to 40% of the added ^{14}C -2,4-D in soils after several weeks of incubation and 50% of the added ^{14}C -2,4-D in finished compost. These results indicate that as in soils, most of the 2,4-D is mineralized or incorporated into NaOH-extractable and humin fractions during composting of yard trimmings.

Composts have relatively high OM contents of 40 to 70% (Table 1) compared with soils, which typically have OM contents of less than 5% (17, 21). This is important, since high levels of OM have been shown to adversely affect the rate of 2,4-D mineralization in soils (10) and it has been suggested that 2,4-D sorbed to OM in soils is not readily available for microbial degradation (20). For example, no significant mineralization of 2,4-D was observed in high-OM soil after 45 days of incubation (10). In contrast to this, our results show that almost 50% of the added 2,4-D was mineralized during the composting of yard trimmings, even though more than 90% of the added 2,4-D was sorbed to compost OM from day 10 onward (Fig. 5). Also, the mineralization of 2,4-D carbon closely paralleled the mineralization of total compost OM to CO_2 (Fig. 3). These results suggest that microbes involved in 2,4-D degradation in compost are different from those in soils in that they are not only able to degrade sorbed 2,4-D but are also able to degrade 2,4-D in the presence of a high level of OM. It is possible that the relatively higher moisture content of the compost, compared with that of soils, led to rapid equilibrium partitioning of 2,4-D between the sorbed and soluble phases, resulting in effective 2,4-D degradation.

Previous studies indicated that mesophilic temperatures of 22 to 26°C are optimal for 2,4-D biodegradation (25). Furthermore, pure cultures of 2,4-D degraders isolated to date are not known to degrade 2,4-D at thermophilic temperatures. The temperature regimen used in this study was designed to mimic the temperatures seen during large-scale composting of yard trimmings in windrows, which range from 55 to 65°C (18, 24). Our results indicate that 2,4-D is mineralized at a composting temperature of 60°C (Fig. 3 and Table 2). When ^{14}C -2,4-D was added to yard trimmings at 2,500 ppm and the composting temperature was maintained at 55°C, 18% was mineralized after 50 days, indicating that the thermophilic 2,4-D-degrading community had a high capacity for 2,4-D degradation (Fig. 6). In future studies, the 2,4-D-degrading thermophilic organisms from compost will be isolated and identified and their 2,4-D metabolism will be compared to that of known 2,4-D-degrading organisms from soils.

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